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Modeling approaches to assess the potential effects of shellfish aquaculture on the marine environment

Approches de modélisation pour évaluer les effets potentiels de la conchyliculture sur le milieu marin

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ABSTRACT

The purpose of this document is to provide advice on the application of mathematical models as tools for assessing the potential effects of shellfish farming operations on the marine environment. Models that provide predictions of the potential effects of aquaculture operations may be used in the planning phase of developments to assess appropriate farm locations and sizes; to provide guidance to all stakeholders on the potential effects of such developments; and incorporated within management decision frameworks to provide objective assessment of potential environmental risk of such developments.

The scale over which the effects of shellfish aquaculture operations may occur, ranging from very localized (within metres) to far-field (kilometers), means that different processes have to be taken into account within models, depending on the type of effect being simulated.

This paper presents three different approaches to modeling specific aspects of the effects of shellfish aquaculture. These range from predicting the near field benthic effects from increased flux of waste material, to effects on lower trophic level ecosystems and finally the use of simple index models of waste output to predict ecosystem level impacts.

We propose that modeling techniques and methodologies are available that simulate and predict shellfish aquaculture-environment interactions with reasonable predictive capability and could already be of use to Habitat Management. As further data become available and validation exercises are completed, predictive skill and confidence in model outputs will increase.

RÉSUMÉ

Le but du présent document est de fournir un avis sur l'utilisation des modèles mathématiques pour évaluer les effets potentiels de la conchyliculture sur le milieu marin. Ces modèles peuvent être utilisés pendant la phase de planification pour choisir les emplacements et les dimensions appropriés des sites de culture. Ces modèles permettent de fournir des conseils aux différents intervenants quant aux effets potentiels de ces activités et peuvent aussi être intégrés aux cadres décisionnels pour évaluer de façon objective les risques pour l'environnement qui pourraient découler de tels aménagements.

Les effets associés aux exploitations conchylicoles peuvent varier d'une échelle très localisée (sur des distances mesurées en mètres) à une très grande échelle (sur des distances mesurées en kilomètres). Pour cette raison, les modèles doivent tenir compte de différents processus, selon le type d'effet simulé.

Ce document présente trois approches différentes pour la modélisation des effets de la conchyliculture. Ces approches incluent la prédiction des effets benthiques à proximité des sites reliées à un plus grand flux de déchets, la prédiction des effets sur les niveaux trophiques inférieurs de l'écosystème et la prédiction des effets à l'échelle de l'écosystème avec des modèles à indice simple.

La simulation et la prédiction des interactions entre la conchyliculture et l'environnement sont possibles grâce à la modélisation. Avec un niveau de prédiction raisonnable, cet outil peut être utilisé par la Gestion de l'Habitat dès maintenant. Au fur et à mesure que l'on disposera de nouvelles données et que des simulations de validation seront effectuées, la capacité de prédiction et le niveau de confiance de ces modèles augmenteront.

INTRODUCTION

The aim of this paper is to provide advice to Habitat Management on the use of mathematical models as tools for assessing the potential environmental effects of shellfish aquaculture operations. This paper is one of five science advisory documents that consider shellfish aquaculture-environment interactions and specifically addresses the following Habitat Management advice request:

> *What modeling methodologies and techniques are available to provide predictions of the potential effect of shellfish aquaculture operations on the marine environment?*

Although the use and development of models in a research context is an advanced science, their application in a regulatory capacity is less well developed (Henderson *et al.*, 2001). Over recent years, modeling effort with regard to shellfish cultivation has focused primarily on predicting and exploiting capacity rather than environmental interactions (although the two are closely linked). With the continuing expansion of the shellfish farming industry, in concert with a worldwide increase in aquaculture operations (Baghen, 1995), there are growing concerns over the potential environmental consequences of shellfish aquaculture operations.

In order to effectively assess the potential environmental effects of shellfish aquaculture operations, Habitat Management require the predictive capability afforded by well defined, parameterized and validated models. Additionally, with the move towards ecosystem based management approaches, modeling tools will become increasingly important in providing advice and guidance on multi-scale (near-field, far-field) assessments of potential effect of different developments.

Henderson *et al.* (2001) list the potential utility of modeling tools applied in the regulation and management of aquaculture operations as:

- Indicators (or warning signs) which might be used for adapting monitoring strategies and against which predetermined standards may be compared
- Descriptors for well-understood physical processes (such as the settling of organic material around fish [and shellfish] farms
- Tools for all sectors to achieve best practice within the process of aquaculture development and its regulation
- A cost effective alternative to extensive field studies that may struggle to differentiate between anthropogenic impacts and the large variations that occur naturally
- A means to derive fast predictions of potential impacts for different aquaculture scenarios
- Contributions to the movement from reactive management to proactive management.

The accuracy of model predictions will be determined by the suitability of the model to the test environment, how the model is configured and the quality of the data used in parameterization. The scale over which the effects of shellfish aquaculture operations may occur, ranging from very localized (within metres) to far field (kilometers), means that different processes have to be taken into account within models, depending on the type of effect being simulated. Ensuring that the model is 'fit for purpose' is of prime importance in the examination of model outputs and developments.

This paper presents three different approaches to modeling specific aspects of the effects of shellfish aquaculture. The first section, "*Modeling the effects of biodeposition from shellfish farms on the near field benthic environment*" *(Chamberlain/Weise)* examines the application of the finfish waste sedimentation model DEPOMOD to predict the benthic effects of suspended mussel farms. The second section, "*Perspectives on the use of mathematical models for assessing environmental effects of bivalve culture*" *(Dowd)* discusses lower trophic level ecosystem models and their adaptation to assess bivalve-ecosystem interactions. And finally, the third section, "*Operational use of simple models in aquaculture management*" *(Grant)* examines simple index models (e.g. spreadsheet-based) of waste output to predict the potential for ecosystem level impacts and their use in culture management. An example with ammonia production and removal is presented with a view towards how these models may be incorporated into the regulatory process.

The main outcome of this paper is to propose that there are modeling approaches available that simulate and predict shellfish aquaculture-environment interactions with reasonable predictive capability. These could already be of benefit and use to Habitat Management. Further research and validation are required on all approaches and at present should be applied cautiously. However, as further data become available and validation exercises are completed, predictive skill and confidence in model outputs will increase.

SECTION 1

MODELING THE EFFECTS OF BIODEPOSITION FROM SHELLFISH FARMS ON THE NEAR FIELD BENTHIC ENVIRONMENT (Chamberlain and Weise)

INTRODUCTION

Commercial cultivation of edible mussels and similar species has been developed in many parts of the world (Mason and Drinkwater, 1981). In comparison to marine cage finfish farming, shellfish cultivation is often considered relatively benign, particularly as there is no addition of organic matter as food sources and medicines/chemotherapeutants are not used. Nevertheless, shellfish are suspension feeders producing fecal pellets and pseudofeces as a waste product from the concentration of organic and inorganic particles within the surrounding seawater. Considerable quantities of this material can be generated by extensive culture sites, where many thousands of bivalves are suspended in the water column, which may accumulate over the relatively large area beneath the farms thereby altering seabed sediment characteristics and the benthic community structure. Jaramillo *et al.* (1992) describe the filter feeding process as the mussels removing fine particles from the water column, repackaging them and returning them to the water as fecal (and pseudofecal) pellets. These pellets, whose settling velocity is greater than that of the original fine particles of which they compose (Simpson, 1982), settle onto the benthos in a process known as biodeposition (Haven and Morales-Alomo, 1966). Thus, through feeding and excretion of bivalves, phytoplankton and fine particles settle out in areas where hydrographic processes would not normally allow this to occur.

Bivalves can be cultured both on and off the bottom using a variety of substrates. A number of different growing and cultivation techniques have been developed around the world and are used within the shellfish production industries. For example, mussels may be cultured on bouchots (poles) driven into intertidal areas, seeded and grown on the seabed in areas of optimal growth conditions or suspended from rafts or longlines supported by buoys. This paper deals specifically with the practice of suspended mussel farming (also referred to as offshore or deep-water culture) where droppers, made of ropes or straps, or continuous mesh socks are suspended in the water column upon which the mussels settle (or are seeded) and grow.

The effects of increased sedimentation, through biodeposition processes, from suspended mussel cultures, on their surrounding benthic environment, have been considered in a number of studies and are reviewed in detail in Mckindsey *et al.* (2006) and Cranford *et al.* (2006). Reported effects vary considerably between studies, with descriptions of the physico-chemical and biological structure of the proximal seabed ranging from no observable effect (Crawford *et al.*, 2003; Danovaro *et al.*, 2004), through slight modifications to the benthic status (Baudinet

et al., 1990; Grant *et al.*, 1995), to highly impacted and enriched conditions (Dahlback and Gunnarson, 1981; Stenton-Dozey *et al.*, 2001). Interestingly, when observed, the location of benthic effect is generally confined to a small area extending no more than a few tens of metres from the farm boundary (Mattson and Linden, 1983; Chamberlain *et al.*, 2001; Hartstein and Rowden, 2004). Hatcher *et al.* (1994) suggested that as mussel feces and pseudofeces are derived from phytoplankton and suspended sediment, they would have similar organic matter to natural sedimentation. Hence, a large volume of mussel biodeposits found beneath a site would represent an increase in total organic deposition driven by a total increase in sedimentation. Consequently, increased sedimentation through biodeposition processes may effectively lead to organic enrichment of the seabed surrounding mussel farms with a subsequent alteration in the physico-chemical and biological status of the proximal seabed conditions.

However, as previously noted, these effects are not always observed – biodeposits from mussel farms thus may, or may not, have significant effects on the benthos.

A number of factors have been suggested to account for the disparate observations of effects of farms on their local environment (Table 1). We consider that these factors may be combined into three broad categories that are characterized by how they influence the potential effect:

- Group A: quantity and quality of material exiting the farm
- Group B: dispersion of material exiting the farm
- Group C: fate of waste material post-deposition

Indeed, Chamberlain *et al.* (2001) considered that the production tonnage of a farm and food availability to stock (Group A) and dispersion of biodeposits from the farm site (Group B) were important factors in determining the final fate of fecal material and any subsequent impact on the benthos. They suggested that current velocity variations could explain the differences in the influence on macrofaunal assemblages reported in other studies. Similarly, Hartstein and Stevens (2005) proposed that given a particular rate of ejection of material into the water column, the rate of arrival per unit area will be strongly a function of hydrodynamic factors serving to spread the material (Group B).

These three categories also coincide with the processes that are simulated and /or parameterized in finfish aquaculture waste particle tracking models (e.g. DEPOMOD, Cromey *et al.*, 2002). These models are of increasing interest and use to Habitat Management and operators alike, in that they provide *a priori* assessment of the potential nature and scale of effect of individual aquaculture operations on their near-field benthic environment.

The impact of particulate wastes from finfish farms is a critical factor influencing the holding capacity of individual farm sites. Although the parallel regulatory framework for shellfish farming is less developed, benthic impact is recognized as

a potential consequence of shellfish farming. It is likely that future regulations will require a more thorough assessment of the environmental impacts of shellfish farming. Mathematical modeling will be central to this and may provide the tools for planning and monitoring as well as regulation to minimize the impacts from aquaculture activities (Henderson *et al., 2001*).

DFO Habitat Management has requested advice on tools and methodologies that will assist them in reviewing shellfish aquaculture site applications and in assessing ongoing aquaculture operations in the marine environment. The objective of this section is to assess whether particle tracking models such as DEPOMOD could be applied in the prediction of near field benthic effects of suspended mussel farms. Essentially, could the fate of particulate wastes from suspended bivalve culture be modeled using similar techniques to those that have been applied to cage finfish farms?

In order to provide advice on this, we present the following:

- A brief overview of waste sedimentation/particle tracking modeling approaches to marine aquaculture operations
- An analysis of the modifications required to apply these models to shellfish farming scenarios
- A case study of the application of DEPOMOD to a shellfish culture site
- A discussion of the benefits and limitations of the presented approach and further work requirements to improve predictive skill

Additionally, different scenarios (e.g. shallow/deep sites, low/high current speeds, low/high biomass) are presented to demonstrate how DEPOMOD may be applied as a management tool to aid mussel growers and regulators in selecting shellfish culture sites, defining site limitations, optimizing production as well as designing and implementing monitoring programs.

SEDIMENTATION MODELING

Modeling the processes and effects of enhanced sedimentation in the marine environment has, to date, concentrated on sewer outfalls, pulpmill effluents and cage finfish farms. These processes are readily amenable to waste sedimentation/particle tracking modeling as the quantity and nature of the material being discharged can be calculated and, to some degree, regulated. A general review of modeling approaches to finfish farming was undertaken by Silvert and Cromey (2000). A number of finfish waste sedimentation models have been developed, Chamberlain *et al.* (2005) presented a detailed analysis of these and examined the potential utility of DEPOMOD (Cromey *et al.*, 2002) in the prediction of benthic effects of large scale marine cage finfish farms in British Columbia. Although identifying limitations in the model and data parameters, Chamberlain *et al.* (2005) demonstrated that such models may be of use in the provision of advice to regulators, habitat management and operators.

By contrast, little attention has been applied to modeling the benthic effects of suspended shellfish farming because, historically, it has been viewed as relatively low risk and not a priority target for model development. However, because of the increasing number and size of commercial shellfish farming operations, often in areas which have traditionally been involved in finfish farming and growing concerns over the number of reports indicating benthic effects from this type of culture, there is an emerging interest in predictive capability similar to that attained for finfish farming.

In overview, finfish farm waste sedimentation/particle tracking models simulate the trajectory of particles (feces and waste feed pellets) as they are released from a cage structure and settle through the water column, providing a prediction of the magnitude and spatial extent of the deposition of material on the seabed. Further predictions, linking the flux of material to changes in the biological structure and/or chemical status of the seabed may be achieved through the use of field measurements and semi-empirical models (e.g. Cromey *et al.*, 2002). The fundamental forcing parameters used in these models were initially reported by Gowen *et al.* (1989) as the hydrographic regime, depth of water and settling velocity of the waste material (Figure 1). To be effective, the overall model must represent all the important processes (e.g. advection, deposition, resuspension etc) that lead to and cause benthic effect.

We consider that these fundamentals will be equally important in determining the fate of biodeposits from shellfish aquaculture sites. The processes acting on the particles as they settle through the water column will be the same and can be simulated through modeling. However, because these models are designed for finfish farms, there are certain aspects of model parameterization and data input that require modification for specific application to the shellfish farm scenario. The importance of good quality and representative data in model parameterization cannot be emphasized enough. Indeed, Chamberlain *et al.* (2005) stress that the accuracy of model predictions will be determined by the suitability of the model to the test environment, how the model is configured, and the quality of the data used.

MODEL PARAMETERS AND SET-UP

The model parameters that require modification in DEPOMOD to be adapted for shellfish farm sites are 1) food loading/excretion rate values as no 'additional' feed is added to the system, 2) sinking rates of mussel fecal and pseudofecal pellets, 3) farm structure configuration to represent mussel lines. These parameters are discussed in detail below, along with the variability and uncertainty associated with these values.

Food loading/Excretion rate

Determining the quantities and type of waste material exiting the farm site per unit time step is a critical component in model parameterization. For the finfish farm scenario, this value may be calculated (within defined envelopes) using the known quantity of food supplied to the farm together with feed conversion and wastage rate estimates. However, biodeposit production rate from mussel farms (both feces and pseudofeces) is not as straightforward and will be dependant on a range of complex and interrelated factors, including:

- Supply of suspended material (concentration and rate)
- Quality of suspended material (organic content, silt content)
- Feeding behaviour of mussels (rate, assimilation)
- Biomass of mussels (stocking density)

which will result in highly variable site (and within site) and temporally specific estimates. Although measuring and/or modeling some of these factors is a relatively simple process (e.g. quantity and rate of food delivered to a farm site), others are not so well understood (e.g. feeding behavior of mussels and hence excretion rates). Indeed, Cranford and Hill (1999) found that daily seston availability and several environmental parameters could only explain 28% of the variation in daily ingestion rates of mussels. With regard to modeling, Chamberlain (2002) encountered difficulties when attempting to apply feed conversion parameters to a known food loading in order to derive fecal and pseudofecal quantity estimates that were environmentally realistic and also produced model results that were representative of field measurements. Consequently, we consider that the application of a generic conversion factor (feed input to biodeposit output) for whole farm scenarios is not possible at present.

An alternate strategy to define the feed load/excretion rate parameter for shellfish farms is to measure and/or calculate the actual quantity of material released from a farm site per unit time step – thus negating the requirement to parameterize and model all the above factors. One approach to calculating this value is to examine the *in situ* biodeposit production rate of mussels at a farm site of interest and extrapolate these values to 'line' (or 'sock') or whole farm scales. Admittedly, there is significant potential for the introduction of errors in such a scaling calculation. However, if the biodeposit production measurements are conducted under actual site conditions, we consider that such estimates and calculations may not be unreasonable. Moreover, when applying such models from a habitat management perspective, it is useful to consider the worse-case scenario situation. Using the maximum biodeposit production rate measured and scaled to a whole farm situation would go some way to simulating such a scenario for the period examined. Consideration of when the peak biodeposit production rate occurs within a grow-out cycle should be made in order to assess overall worse case scenario estimates.

The second approach described above is applied in the presented case study using biodeposit production rates from the farm site examined. As noted, both of the above approaches to estimating the feed load/excretion rate values for shellfish culture sites have considerable limitations and assumptions. Further research is necessary to better define this parameter.

Sinking Velocity

The particle sinking rate is a dominant parameter requiring characterization in all particle tracking models. The sinking rate of particles depends on several properties of the particles including size, density and porosity and is often based on variations of Stokes Law (Hendricks and Eganhouse, 1992; Cromey *et al.*, 1998). However, Stokes Law assumes that particle density and shape are similar across all size ranges – not an assumption one can make with fecal particles. Therefore, to obtain estimates of the settling velocity of these particles, a number of studies have calculated this value through experimental means.

Data on the sinking velocity of fecal pellets produced by bivalves is relatively scarce. This has only recently been addressed in some studies. For the mussel *Mytilus edulis* Chamberlain (2002) reported fecal sinking velocities of less than 0.5 cm s^{-1} for 4 cm individuals. Callier *et al.* (2006) measured sinking velocities in the range of 0.3 to 1.8 cm s^{-1} for mussels in the 3 to 7 cm size class (shell length). Sinking velocities of 0.1 to 4.5 cm s^{-1} have been reported for the mussel *Perna canaliculus* (Giles and Pilditch, 2004; Hartstein and Stevens, 2005) and 1.1 to 3.0 cm s^{-1} for *A. zelandica* fecal pellets (Miller *et al.*, 2002). The range of sinking velocities encountered in these studies is likely due to the size and composition of fecal pellets. Several studies have found that larger mussels produce larger faecal pellets which sink more rapidly (Giles and Pilditch, 2004; Callier *et al.*, 2006). Other studies have shown that food quality influences fecal pellet density and hence settling velocity. For example, mussels fed on diets with high silt content produced faecal pellets that sank more rapidly than those produced from algal diets (Chamberlain, 2002; Miller *et al.*, 2002; Giles and Pilditch, 2004). Finally, bivalves may produce pseudofeces (particles that are not ingested) under certain environmental conditions such as high SPM concentrations. Because of their fluffy texture, these pseudofeces tend to have slower sinking velocities than fecal pellets (Giles and Pilditch, 2004) and may thus be dispersed even further than faecal pellets. These different particle types and their sinking velocities can be modeled in DEPOMOD.

The biodeposits produced at shellfish culture sites are expected to disperse farther from their origin than those produced at finfish culture sites because of their slower sinking velocities. Of course the area over which biodeposits will settle on the benthos will be strongly influenced by water movement and depth of site (i.e. residence time in the water column).

Cage set-up/Particle release location

Most particle tracking models require the location of the particles' start position to be defined. For the finfish farming scenario, this generally involves defining the location and dimensions of the cages, after which the particles are then released at random points within the structure.

For the suspended shellfish culture scenario, the 'cage' may be defined as the whole farm, individual lines or single droppers. The selection of which of the above will be dependant on the scale of assessment, number of particles modeled and computer processing time.

CASE STUDY

The following section describes the application of DEPOMOD to a suspended mussel culture site in Great Entry Lagoon (GEL), Magdalen Islands, Quebec. The model outputs of solids deposition are compared with field measurements (described below) of sedimentation to assess model performance and predictive capability.

The general layout of the mussel culture site at GEL is illustrated in Figure 2. The culture site covers a 2.5 km^2 area and is located in approximately 6.5 m of water. The farm produced 180 ton yr^{-1} and had been in operation since the 1980's. The mussels were cultured in a 2 year grow out cycle (0+ and 1+ mussel cohorts) on a long-line system from which individual mussel socks or continuous mesh sleeves were suspended.

Model input parameters (Table 2) are based on *in situ* measurements of biodeposit production rate, fecal pellet settling rate and hydrographic data. A model grid of 499 x 499 cells with a fine grid cell resolution (1 m) was used for all model simulations. In this exercise, we chose to model specific individual lines (9 backlines) rather than the whole farm since we expected limited dispersion of biodeposits as the mussel sleeves were close to, or in some cases touching, the seabed during the study period. Thus a 'cage' is represented by a single mussel backline.

In this study, we use excretion rate values based on the *in situ* biodeposit production rates measured by Callier *et al.* (2006) at the GEL site. Maximum biodeposit production recorded was 125.6 mg d^{-1} ind^{-1} from a batch of 1+ mussels. Using this maximum value as the potential worse case scenario, we extrapolated this measurement to calculate the potential biodeposit production rate from a mussel line as 26.4 kg $line^{-1}$ day^{-1} (365.8 m length; stocking density 575 mussels m^{-1}). This loading was split into 17.6 kg feces and 8.8 kg pseudofeces using a 67:33 feces to pseudofeces production ratio based on the calculations of Bayne *et al.* (1993) in similar ambient seston concentrations. Based on data from Callier *et al.* (2006), a sinking speed of 1.0 ± 0.3 cm s^{-1} was used for faecal pellets. We attributed a sinking speed of 0.2 ± 0.02 cm s^{-1} for

pseudofeces based on observations by Walker *et al.* (2005) for flocculent material that was considered to be pseudofeces.

Field measurements of sedimentation

Model outputs were compared with field measurements of sedimentation. The sedimentation rates were evaluated using paired sediment traps (separated by 4 m) positioned at distances of 0, 3, 6, 12, 15 and 30 m along transects perpendicular to the edge of the mussel farm and to the mussel lines themselves (towards the SW, SE and NW – Figure 2). Following a 24 hour deployment, the traps were retrieved and the contents were filtered and weighed as described in Callier *et al.* (2006).

Model Outputs of flux

The predicted solids flux from the mussel lines (Figure 3a&b – units in g solids m^{-2} d^{-1}; darker colours indicate higher predicted flux) shows that the main area of deposition was directly beneath the farm site, a reflection of the very shallow nature of the site. Predicted flux reached values close to 40 g m^{-2} d^{-1} and the spatial extent of deposition was greatest towards the south-west (15 m) corresponding to the direction of the major current flow during the sampling period (Figure 3c) (Average 7.1 ± 3.6 cm s^{-1}, towards the SW). The model predicted limited dispersion towards the north-west (3 m) and south east (9 m).

Predictions of solids flux were extracted from the model grid at each of the locations of the sediment trap pairs. Comparisons between predicted and measured flux are presented in Figure 4a. When adjustments were made to account for natural background sedimentation, the model predictions compared favourably with the observed flux as illustrated in Figure 4b. The model, however, tended to underpredict the scale of sedimentation when compared to field data measurements. It is uncertain at present whether this was as a result of the model (configuration, parameterization, process) or field analysis (capture device artefact).

DISCUSSION

Overall, the model performed reasonably well, with predictions of increased solids flux coinciding with similar measurements. Although the data is scattered, an observable trend in measured against predicted flux can be seen. The preliminary results presented here are part of a comprehensive study that was undertaken between 2003 and 2005 at three mussel culture sites in Quebec (GEL described above, Harbour House Lagoon also in the Magdelan Islands and Cascapedia Bay in Chaleur Bay). Extensive field data were collected to characterize the seabed conditions along biodeposition gradients at these sites (including sedimentation rate, macrofaunal benthic community, sediment chemistry). These data will allow extensive analyses of model outputs, and comparisons with a range of actual field measurements. Sensitivity analyses will be carried out to examine the relative

importance of each component of the model parameterization which will further improve model performance and predictive capability.

The above case study is an improvement on Chamberlain (2002), in which techniques to apply DEPOMOD to shellfish farming scenarios were first explored. In their study, the model was applied at three mussel farm sites (1 in Scotland, 2 in Ireland) and the outputs, quantified as predictions of Infaunal Trophic Index (ITI) (Word *et al.*, 1978) values from a semi-empirical model validated for finfish farms in Scotland (a sub-module within DEPOMOD) were compared with field measurements. As previously noted, difficulties were encountered in quantifying the feed load/excretion rate values. Chamberlain (2002) found that the model tended to overpredict the degree of impact when food loading and feed conversion estimates were applied, whereas estimations of the quantity of material exiting the farm structures resulted in underestimates of effect. Chamberlain (2002) concluded that there were processes involved that were not taken into account in the modeling process. Thus further research is necessary before realistic predictive capability can be attained.

The modeling approach presented by Hartstein and Stevens (2005) did not make predictions of the mass of material depositing on the seabed, making comparisons with the presented case study difficult. However, their approach of examining the sensitivity of model predictions to a range of diffusion coefficients provides a useful example of the type of analysis that will be required for future DEPOMOD development.

There are a number of other studies currently ongoing to examine the application of similar modeling techniques to predict the effect of increased sedimentation from suspended shellfish farms on the benthic environment. These include the two additional sites in Quebec noted above (www.dfo-mpo.gc.ca/science/aquaculture/acrdp-pcrda/quebec/Q-03-01-001_e.htm) and the EU funded Framework 6 RTD project ECASA (Ecosystem Approach to Sustainable Aquaculture – www.ecasa.org.uk). As further outcomes and information becomes available from these projects, confidence in the predictive capability of such models will increase.

All of the above have demonstrated (are demonstrating) the potential utility of waste sedimentation models as useful tools for assessing the potential effect of suspended shellfish farming. It is important to note that such models may also be used to identify when effects are *not* likely and where there is little risk to the benthic environment. One key element that is currently not well parameterized is the linkage between increased flux of shellfish bioproducts (measured as solids and/or carbon) and changes to the benthic status. For waste sedimentation/particle tracking models to be of 'effective' use to Habitat Management, it is essential that such coupling of pelagic and benthic processes is well described and defined within known envelopes of uncertainty/error. Preliminary field trials examining the dose (biodeposit flux) – response (benthic

community index/geochemical measure) relationship have already been conducted to assess the effect of a range of biodeposition rates on seabed conditions (Callier *et al.*, 2004). As further data become available, this will certainly improve model utility towards management decision frameworks.

Another potential benefit in the application of waste sedimentation models is as a management tool to aid mussel growers and regulators in selecting shellfish culture sites, defining site limitations, optimizing production as well as designing and implementing monitoring programs. As an example, here we present the model outputs from a range of shellfish farming scenarios to demonstrate how differences in site water depths, stocking densities and hydrographic regimes can affect the overall footprint of potential farm sites.

The same input parameters as in the above case study (Table 2) were applied except for those parameters that were being evaluated (depth, stocking density and current speed). The model outputs (Figures 5 - 7) show predicted solids flux (g m^{-2} d^{-1}) on the seabed around the mussel lines.

Shallow site versus deep site
Scenario 1 (Figure 5) illustrates the effect between a mussel culture site located in a shallow site (4 m) and a deep site (20 m) and compares the flux of solids deposition. A feed input of 26.4 kg $cage^{-1}$ d^{-1} and an average current speed of 2 ± 1 cm s^{-1} were used for both simulations. The contour plot indicates that solids flux is greater and more constrained at the shallow site whereas particles are dispersed further (larger footprint) and at a lower flux at the deeper site.

Low and high stocking density
Scenario 2 (Figure 6) shows the effect of different stocking densities on the predicted depositional footprint (26.4 kg $line^{-1}$ day^{-1} and 52.8 kg $line^{-1}$ day^{-1}). A depth of 4 m and an average current speed of 2 ± 1 cm s^{-1} were used for both simulations. As expected, the resulting model predictions show a higher level of predicted flux beneath the mussel lines with high stocking density.

Depositional versus dispersive site
Scenario 3 (Figure 7) demonstrates the predicted effect of locating a mussel culture site in a depositional versus more dispersive site. For both simulations, a depth of 4 m was used and a feed input of 26.4 kg $cage^{-1}$ d^{-1}. For the depositional site, an average current speed of 2 ± 1 cm s^{-1} was used. The current was increased five-fold, averaging 10 ± 4 cm s^{-1}, to represent a dispersive site. Model predictions show a lower flux and greater dispersion at the dispersive site.

CONCLUSIONS AND RECOMMENDATIONS

The application of DEPOMOD at mussel culture sites in the Magdelan Islands, Scotland and Ireland is clearly promising with regard to predictive capability. Although predicting the benthic effect of suspended shellfish farming is at an early

stage of development, the model outcomes presented demonstrate the potential utility of such modeling techniques. It is apparent that the physical processes affecting the deposition of particulate waste from shellfish sites are reasonably well parameterized in the model. However, at present, we consider that the data gaps and uncertainty surrounding other key model parameters are limiting to the general use of these models and outputs by Habitat Management.

By far the most important area of uncertainty is the feed load/excretion rate parameter. This is a critical input into the model as it will affect all subsequent predictions which would form the basis of any management decisions. Although there are emerging and promising modeling techniques to better define this input parameter, the application of a generic conversion factor (food input to biodeposit output) for whole farm scales is still premature. As new data become available for all of the different parameters, confidence in model predictions will be increased. Similarly, coupling biodeposit flux and changes to the benthic status (biology/chemistry) with linkage to model outputs will improve model performance and the ability to assess predictive capability. As higher levels of validation are achieved, this will provide useful information for incorporation into decision making frameworks.

It must be kept in mind that all models have assumptions and limitations because they attempt to simplify the processes they attempt to reproduce. There are a number of processes that are not been considered in this modeling exercise that could potentially have a significant effects on the fate and impact of biodeposits. These include for example the effects of 1) reduced water movement due to the physical structure of the farm, 2) post-deposition modification of particles (resuspension, degradation), 3) the application of a spatially homogenous horizontal flow field within the model domain, and 4) turbulence on the integrity of fecal and pseudofecal particles. The relative importance of these and other factors should be examined in future research programs.

We expect that ongoing research on all of these different issues will result in significant improvements in model performance. Further testing of the model at additional sites in a range of environmental conditions will be necessary for an acceptable level of validation. Thereafter, the general application of the model may be applied through sensitivity analysis.

SECTION 2

PERSPECTIVES ON THE USE OF MATHEMATICAL MODELS FOR ASSESSING ENVIRONMENTAL EFFECTS OF BIVALVE AQUACULTURE (Dowd)

The aim of this section is to introduce and explain a generic mathematical modeling framework for describing the lower trophic levels of a coastal ecosystem modeling, as well as to discuss its adaptation to include bivalve aquaculture. The goal is to show how this quantitative tool can be used to assess the extent to which bivalves interact with and affect their supporting ecosystem. An application (to Tracadie Bay, PEI) provides a concrete illustration of these ideas. New directions in integrating models and measurements in order to improve predictive skill are also discussed. Finally, some remarks are made on the usefulness of such mathematical models from a management perspective.

OVERVIEW OF LOWER TROPHIC LEVEL COASTAL MARINE ECOSYSTEM MODELS

Bivalve aquaculture relies on, and interacts with, its supporting ecosystem. Hence it is natural to consider the mathematical modeling of ecosystems as a tool for quantifying the environmental and ecosystem effects of intensive bivalve culture. In this section, a general and widely used approach to the biogeochemical modeling of coastal marine ecosystems is outlined and adapted to include bivalve aquaculture. The emphasis is on lower trophic levels and both pelagic and benthic components are included. Please also note that the class of models considered here describe the properties of ensembles of individuals, generally measured in concentrations units (an Eulerian perspective). An alternative framework in which one follows individuals and quantifies their interactions is possible – such Lagrangian "individual based models" are not considered here (see Woods, 2005)

The state variables that describe the pelagic ecosystem components typically include the following groups: plankton, nutrients, and detrital organic matter. Depending on the level of complexity desired, each of these groups can be aggregated in various ways. For instance, it is common to consider plankton in terms of functional groups (e.g. diatoms, dinoflagellates, macrozooplankton), but often just a single composite plankton group (e.g. the phytoplankton) is used. For the inorganic nutrients, different elements (e.g. nitrogen or phosphorous) can be considered in their various chemical forms, or a single most limiting nutrient might be used. Higher trophic levels (beyond secondary production) are generally not considered explicitly, and are accounted for by loss term which provides for ecosystem closure (Edwards *et al.*, 2001). Fennell and Neumann (2004) provide a comprehensive review of these considerations, as well as of the current state of the art in marine ecosystem modelling.

Consider the schematic diagram of a coastal ecosystem given in Figure 8. The pelagic ecosystem state variables are the standing stocks of phytoplankton (*P*), nutrients (*N*) and detritus (*D*) (note that, for simplicity, zooplankton are not being considered). Interactions between state variables are indicated by arrows that represent fluxes. These have to mathematically specified as a function of the ecosystem state and environmental conditions and drive the cycling of matter (and energy) in the system. The major pelagic ecosystem processes to be considered include:

- *Primary production*: the conversion of *N* to *P* through autotrophic processes
- *Predator-prey interaction*: the grazing of *P* by higher trophic levels
- *Remineralization*: the conversion of organic *D* to inorganic *N*

Other biogeochemical and ecosystem processes such as mortality and sinking are also included. Since these pelagic components are embedded in a fluid environment, advection and mixing must also be considered. A guiding principle in the construction of such models is the purpose for which they are being used. In this instance, the focus is on the interaction of bivalve culture with its supporting ecosystem, and so the ecosystem description is kept as simple as possible.

In shallow coastal systems, the benthos has a strong two-way coupling with the pelagic ecosystem. Major benthic ecological processes include:

- *Sedimentation*: sinking of water column *D* (and *P*).
- *Resuspension*: re-injection of settled organic matter into the water column.
- *Burial*: permanent removal of organic matter to the sediments
- *Remineralization*: the *N* efflux resulting from sediment biogeochemistry.

Benthic state variables are distinguished from the pelagic counterparts above in that they are spatially fixed. Once again, an important choice here is the level of aggregation and complexity required, considered in light of the goals of the study. Full diagenetic models of sediment biogeochemical processes are available (Boudreau, 1997) and Soetaert *et al.* (2000) outlines a set of useful approximations to these complex models. For the purposes of adapting ecosystem models for bivalve culture, a level of intermediate complexity seems sensible. The schematic of Figure 8 outlines such a case wherein a single benthic state variable – the detrital pool, *B* – receives organic matter inputs from the pelagic system through the net of sinking and resuspension of *D* and bivalve biodeposition. Its output is an *N* efflux associated with benthic remineralization.

Mathematically, the conceptual diagram of Figure 8 can be translated into a system of nonlinear, coupled ordinary differential equations. Define *X* as the vector containing the values for the ecosystem state variables (here *P*, *N*, *D* and *B*). The ecosystem model can be represented as

$$\frac{dX}{dt} = f(X,\theta) + g(t) \qquad (1)$$

where the left-hand-side of (1) is the time rate of change of the ecosystem state variables. Interactions between and within ecosystem components are embodied in the nonlinear vector operator $f(\bullet)$, which is functionally dependent on the current state of the ecosystem, X, and on a set of parameters, θ. External forcing, or sources and sinks, are represented by $g(t)$. Such equations are readily solved using standard numerical integration methods (e.g. Runge-Kutta) to yield the time evolution of the ecosystem components $X(t)$.

The class of models described above provides a basic description of an ecosystem comprised of interacting populations and contained within a finite volume. To incorporate a spatial dimension, models of the form (1) can be coupled together in a box modeling framework (e.g. Raillard and Menesguen, 1994). Note also that it is often desirable to add special feature like delays (e.g. a benthic N efflux based on a lagged time history of the sedimentation) and stochasticity (e.g. resuspension events of variable duration and magnitude) (c.f. Dowd, 2005).

In order to properly embed such models to include the spatial and temporal effects of currents and mixing processes on the redistribution of ecosystem state variables it is necessary to solve a system of partial differential equations of the form

$$\frac{\partial X_i}{\partial t} + \vec{u} \cdot \nabla X_i - \nabla \cdot K \nabla X_i = f_i(X_1,....,X_n,\theta) + g(\vec{x},t), \quad i = 1,....,n \qquad (2)$$

where X_i is the ith ecosystem component, $\vec{u}$ is the velocity field, and K is a dispersion coefficient. The left-hand-side of the equation represents the movements and mixing of freely floating ecosystem components (note that the benthic component would be fixed in space). This set of non-conservative tracer equations would correspond to the ecosystem equations in (1) – with the operators f and g defined analogously - coupled to an ocean circulation model. Such bio-physical models are of much current interest for understanding ocean ecosystems (Hofmann and Lascara, 1998) and also being used to assess aquaculture ecosystems (Duarte *et al.*, 2001; Dowd, 2003)

While the above modeling framework is straightforward in its conceptualization, there are additional important issues for application. These arise primarily from two main features: (i) the models describe population interactions and are fundamentally nonlinear, and (ii) the functional form of the governing equations are unknown or uncertain, since there are no "ecological first principles" from which they can be derived. The result is that the seemingly sensible notion of adding more ecological processes and state variables to reflect our ecological knowledge does not necessarily lead to improved predictive skill. A major decision for the investigator is thus the level of complexity desired. This must be considered in

light of the goals of the study, the ability to understand and interpret the nonlinear dynamical behaviour, and by the data available for validation ("Everything should be made as simple as possible, but not simpler" – Albert Einstein.)

INCORPORATING BIVALVE AQUACULTURE INTO ECOSYSTEM MODELS TO ASSESS ENVIRONMENTAL EFFECTS

It is relatively straightforward to incorporate a population of grazing bivalves within the ecosystem modeling framework outlined above. The bivalve component is unique in that while it is a pelagic ecosystem component (in suspended culture), it is also fixed in space. Denoting the bivalve state variable as *M* (for mussels), means that we can add another equation to the model which, in words, takes the form,

$$\frac{dM}{dt} = [\text{ingestion-excretion-feces}]\text{-}[\text{harvest}]$$

This states that the rate of change of the mussel population biomass is a function of the mussel mass balance. The first term on the right-hand-side is the just the net of gains (through the ingestion of *P* and *D*) and losses (through fecal production biodeposited to *B*, and through excretion into the *N* pool). The second term is the amount of the population biomass that is removed through harvesting activities. A full eco-physiological model of bivalves would describe this energy balance as a function of the levels of ecosystem state variables (*P* and *D*) as well as in terms of environmental parameters such as temperature and taking account of the physiological state of the bivalve (e.g. Raillard and Menesguen, 1994; Dowd, 1997). However, considerable simplification can be realized by treating it as a diagnostic variable (Chapelle *et al.*, 2000) by prescribing *dM/dt*=0 - further details on this are found below.

Some results from an application are presented to make more concrete the idea of using an ecosystem model to assess bivalve aquaculture effects. The application is to Tracadie Bay, P.E.I. and follows Dowd (2005) and Strain *et al.* (*in prep*). Tracadie Bay (shown in Figure 9) is a shallow semi-enclosed coastal embayment that supports an intensive mussel aquaculture industry. An extensive multi-year field program has been carried out here and the model application outlined below was calibrated with these field data. The influence of bivalve culture on ecosystem dynamics is considered from the perspective of both its effect on standing stocks, as well as on nutrient cycling.

The model structure follows the generic form given in Figure 8. It is comprised of a *PND* pelagic model with a benthic detrital pool *B*. A population of grazing bivalves is superimposed on the ecosystem. These are not modeled dynamically, but assumed to be in a steady state (*dM/dt=0*) wherein population growth is balanced by removal through harvesting activities. This allows for ecosystem effects to be assessed, but avoids modeling the details of bivalve physiology (while these are

well understood, they are characterized by strongly nonlinear relations which lead to parameter sensitivity and a degradation of predictive skill, c.f. Dowd (1997)). The bay was divided into three spatial zones (boxes) as indicated in Figure 9. Pelagic ecosystem components are exchanged between zones, and with the offshore. Mussel aquaculture is concentrated in Boxes 1 and 3.

Figure 10 shows the model predicted annual cycle of the ecosystem state variables *P*, *N*, and *D*. Nutrients and phytoplankton are seen to be high in spring and fall, and the depletion of nutrients in summer depresses the *P* biomass. The stochastic formulation of the resuspension process is evident in the fluctuating *D* time series. During winter, the internal biology of the bay is effectively turned off (with low temperature and light levels) and the system reverts to the far field values. Spatial variations are also evident with Box 2 being distinctly different from the others. The reasons are twofold: (i) the large spring nutrient input into Box 2 (Winter Bay) due to freshwater runoff input leading to elevated nutrients, and (ii) the lack of *M* grazing of *P* and *D* in Box 2.

Figure 11 shows the simulated annual cycle of the benthic detrital pool *B*. In Boxes 1 and 3, where the M population is found, there is an accumulation of *D* in the spring and early summer as water column primary production is diverted to the benthos by mussel biodepostion. After this time, *B* decays with increased resuspension and remineralization (and the system moves from net autotropic to net heterotropic). In Box 2, there is no mussel biodeposition and hence no accumulation of *B* with a rough balance between episodic resuspension and natural sedimentation through settling.

In addition to the reporting on the standing stock of ecosystem state variables, the ecosystem fluxes (which correspond to the arrows in Figure 8) can also be determined. The first column of Table 3 reports these as annual totals for the baseline simulation being discussed. These results indicate that ecosystem fluxes due to mussel grazing are comparable to other non-bivalve related terms; the most notable factor being their significant role in increasing benthic-pelagic coupling (i.e. the $M \rightarrow B$ flux).

A central question of this section is: how might one use such a mathematical description of an ecological system to assess environmental effects of bivalve culture? The simulation results presented here describes the role of mussels in nutrient cycling in the ecosystem. Further numerical experiments can be carried out to look at other scenarios (with the caution that such scenarios are unrealizable in practice, and predictions cannot be validated with data). Such an exercise was carried out wherein the total mussel biomass in the bay is set to zero, but the system is otherwise identical. The results are shown in column 2 of Table 3, and the third column of differences reporting the difference between the cases with mussels present and mussels absent. The general conclusion is that ecosystem production is diverted to bivalves leading to increased organic loading of the benthos. Direct removal of mussels through harvest is a relatively small effect.

The aquaculture system relies more heavily on (imported) offshore production of P and D, but also uses up more of the *N* inputs into the bay from land runoff.

REMARKS ON USING OBSERVATIONS WITH MATHEMATICAL ECOSYSTEM MODELS

This section has so far outlined how mathematical models of aquaculture-ecosystems can be formulated and applied, as well as how they can be targeted at issues such as environmental impact. However, such studies are not complete until the information from models and field observations are systematically integrated and compared. Below, some of the ways in which data enters various steps in the model building procedure are outlined:

1. *Model forcing*: Field observations provide basic input information needed for model integration (the *g* function in (1)). For the application here, forcing functions included observed annual cycles of light, temperature, as well as the far-field concentrations of the ecosystem state variables.

2. *Model parameters*: Model parameters, θ, are also required to run the model. Values for these can be taken from the literature, but others must be determined on a site-specific basis. For example, for the Tracadie Bay case, both mixing and mussel mass balance parameters balance were determined using field data.

3. *Model calibration*: Model predictions of ecosystem state variables must be compared against measurements. Some predicted quantities will be directly observable; others can only be indirectly measured. Model tuning then proceeds as the iterative refinement of the model, so predictions better match observations. This is typically done through the systematic alteration of important, but poorly known, parameters or forcing functions (while keeping them within reasonable bounds).

The above elements comprise a traditional approach to model building. The primary purpose of the resulting mathematical models resulting from these procedures is a process oriented understanding of bivalve-ecosystem interactions. However, in order to focus on the problem of predictive skill, measurement information must be more systematically integrated with model dynamics.

New approaches are being considered in the ecological modeling community to make more effective use of observations. Of particular interest are the new types of ecological observations with complex spatial and temporal structure obtained from technologies such as remote sensing and moving underwater measurement platforms. New directions in modeling relevant to improving predictive skill of aquaculture-ecosystem models include:

1. *Data assimilation*: This refers to the use of dynamical ecosystem models together with available observations in order to improve estimation of the ecosystem state variables, and the parameters. Methods are based on statistical estimation and are generally split into two categories: (i) retrospective (hindcasting) applications based on calibration via parameter estimation (Vallino, 2000; Dowd and Meyer, 2003), and (ii) nowcasting and forecasting based on state space models (Dowd, 2006).

2. *Model validation*: a proper validation exercise splits the observations into two sets: one used for "training" the model (calibration), and another that is withheld and used only to assess model performance. Unfortunately, this is often difficult in practice due to the data paucity of most ecological studies (under-sampling with respect the time and space scales that must be resolved).

3. *System identification*: Ecosystem equations are uncertain and an appropriate set of mathematical equations must be identified by the investigator. Generally, the level of complexity is chosen qualitatively - based on the goals of the study and the confidence in our ability to model (or parameterize) the ecological processes. However, more formal statistical model building techniques can be applied to compare the performance of two or more competing models, or to decide whether additional complexity is warranted (e.g. Carlin and Chib, 1995).

4. *Adaptive Sampling*: Models can provide guidance for the design of field sampling strategies. Such observing system simulation experiments are being used in geophysics, oceanography and atmospheric science. They use model dynamics and statistical properties of the measurements to maximize the information content of a sampling protocol.

5. *Stochastic models*: Many observed ecological processes are naturally described in terms of their statistical properties (e.g. episodic resuspension). Measurement based stochastic descriptions of parameters or forcing which have important consequences for model predictability (Bailey *et al.*, 2004).

CONCLUSIONS – POTENTIAL APPLICATION IN A MANAGEMENT FRAMEWORK

Marine ecosystem models have made many advances in recent years and are being widely applied in coastal marine systems. A well established mathematical modeling framework exists for the lower trophic levels of marine ecosystems, and is readily adapted to include bivalve aquaculture activities. Formulation and application of such models must be done on a site-specific basis. Present models are targeted at a process-oriented understanding of bivalve-aquaculture ecosystems. Improving the predictive skill of these models requires systematic comparison and integration of measurements. Some remarks on the use of

mathematical ecosystem models relevant to coastal ecosystem management are outlined below.

Pros

1. A carefully designed ecosystem model and supporting measurement program can provide an understanding of how a coastal ecosystems works, and under what conditions significant aquaculture impacts might be expected.
2. Ecosystem models are the only quantitative tool available for scenario analysis, such as looking at the effect of different levels of bivalve culture on ecosystem processes.
3. A robust and carefully tested model can, with some effort (reformulation and recalibration), be transportable between sites.

Cons

1. Ecological models are an emerging, but not yet mature, technology. Current models emphasize a process-oriented understanding of the system under consideration. The way forward involves using new observing technologies and systematic incorporation of these data into models with an emphasis on predictive skill. This is an area of active research.
2. Ecosystem models require careful application by trained practitioners. They also require a long term commitment, continual refinement, and application and testing on a variety of cases if they are to be useful.
3. The data requirements for a proper model validation are significant (and scale with the complexity of the model) often requiring targeted multi-year field programs.
4. The level of confidence that we have in ecological predictions from aquaculture-ecosystem models varies widely. Different formulations of these nonlinear models may have a dramatic and unforeseen effect on predictive skill. A systematic assessment of robustness and confidence in predictions is needed.
5. Complexity in an ecosystem model does not lead to better predictions. A more complex model includes more processes, but also leads to more complicated dynamical behaviour.

In summary, the use of mathematical ecosystem models for coastal management purposes is taking place worldwide. Such models provide for a quantitative synthesis of our ecological knowledge of how coastal systems work. They therefore allow for a systematic assessment of ecosystem-aquaculture interactions, as well as predictions about the effect of coastal development scenarios (such as the introduction or expansion of bivalve aquaculture activity). However, ecosystem models are still primarily research tools with the central aim of better understanding complex coastal systems. In contrast, management needs require robust predictive tools that work in a transparent and easily understandable

manner. Towards this end, a promising avenue for achieving management goals would be to develop even more highly simplified and robust (non-dynamical) models (see Section 3). These, in fact, would be derived directly from the types of mathematical models currently used for ecosystem simulation and described here. The idea is to identify dominant processes and key observables quantities (such as flushing, phytoplankton growth, mussel filtration), and then to combine these variables into simple, easily understandable and robust calculations suitable for management needs.

SECTION 3

OPERATIONAL USE OF SIMPLE MODELS IN AQUACULTURE MANAGEMENT (Grant)

INTRODUCTION

The consequences of sea-based aquaculture for the marine environment involve two fundamental questions:

1. How does the coastal environment influence the growth and survival of the cultured organisms?
2. How does the culture activity affect the food chain and habitat of the ambient waters?

The first question is more related to economic concern of the farmer rather than environmental regulation. Nonetheless, it is a reasonable assertion that cultured animals will only thrive in a healthy environment and to this extent the answers to both questions is of concern to everyone. I will however focus on question #2 because it is the emphasis of the present needs of Habitat Management.

An expansion of question #2 leads immediately to the contrast between near-field and far-field effects, and thereby local versus ecosystem impacts. The intended role of modelling in providing answers to this question is ostensibly simple: in selecting new sites, or expanding existing sites, how much culture is too much? Unfortunately, more questions immediately follow: what is the measure of 'too much', and how can it be predicted in advance of harm to the environment? In this capacity, the role of modelling is precautionary, and intended to guide culture activities toward sustainability.

In this section of this modelling paper, I have chosen not to write in conventional paper format, instead concentrating on practical concerns for applying models to aquaculture management. This excludes a detailed review of approaches and references in favour of a conceptual and realistic 'how to' guide.

MODELLING OBJECTIVES

The first stage in approaching these issues is to define habitat/water/sediment quality goals, collectively referred to as EQO (environmental quality objectives). Ideally, these quantities could be predicted in advance of culture, and then measured directly as monitoring variables to validate the predictions. It is however necessary to backup to a more general level of query and ask, 'about which problems arising from culture should we be concerned?' One answer to this question is organic loading of particulates to the sediments arising from fecal waste. Suspension-feeding bivalves concentrate phytoplankton and other particles from considerable volumes of water relative to their culture area and turn them into

denser biodeposits with greater potential for local sedimentation. The provision of this food supplement to the benthos may cause higher oxygen consumption than the rate at which oxygen is renewed, leading to suboxic conditions. Once oxygen is depleted, benthic decomposition by bacteria proceeds *via* sulfate reduction leading to loss of biodiversity.

Given this scenario, maintaining the status quo of benthic oxygen conditions is a reasonable EQO. However, direct measurement of oxygen in sediments is not routine for many technical and practical reasons, and we look toward indirect and more practical indicators. Sulfide and redox fill these roles, but are less amenable to modelling in an operational context due to their reliance on diagenetic models. For simple models, one might regress a step to a more manageable target, namely, evaluation of organic deposition that caused the oxygen stress (Grant *et al.*, 2005).

Nutrients also receive a lot of attention as an aquaculture impact, particularly ammonia which is excreted by the cultured animals and regenerated from sediments receiving fecal waste. Elevated levels of ammonia can be detected at shellfish culture sites (Hatcher *et al.*, 1994), but as with any water column quantity, there is extreme variation in concentrations due to tide, depth, season, etc. This spatial and temporal range makes it less easy to establish as a monitoring variable, and the short-term consequences of elevated ammonia are hard to detect. Water quality guidelines for ammonia are high relative to typically observed values (e.g. http://www.env.gov.bc.ca/wat/wq/BCguidelines/ ammonia.html#tab1). Nonetheless, ammonia is a waste product whose production by fauna and removal by tides can easily be estimated, so it meets the criteria for a useful index model.

WHAT IS FEASIBLE?

Following this example, a useful exercise is to consider the range of effects cascading from elevated ammonia levels, and the corresponding capability of models and measurements to detect ecosystem responses. This is expressed in the tabular graphic of Figure 12. Although the example is for dissolved wastes, a similar case could be made for particulate wastes. I begin with the source of the waste product and carry through to its expression as an impact. The potential for models and measurement is assessed at each step.

Considering the source of waste material, the rate of production of ammonia from fish or shellfish is known from bioenergetics calculations and can readily be modelled, but its measurement is technically complex, and not appropriate for monitoring. In the water column, those wastes can easily be detected as they disperse, and this loss can be estimated *via* exchange as described below. Aside from toxicity to the cultured animals, the consequences of elevated ammonia are purported to be excess growth of nuisance macroalgae such as *Ulva* (Robinson *et al.*, 2005). This is readily observed in the intertidal, but modelling is problematic because its link to excess nutrients from aquaculture is tentative. It has been

examined in more sophisticated models (Bergamasco and Zago, 1999). A further effect of nutrient input may be the occurrence of harmful microalgal blooms, but again a cause-effect with aquaculture waste is unsubstantiated, or at least not suitable for operational modelling in culture management. Concerning measurements, detection of HAB is a regular part of some monitoring programs, but is otherwise an expensive proposition. Finally, excess nutrients which cause phytoplankton blooms may promote increased sedimentation, a classic eutrophication response. This can be measured via sediment traps, but models of sedimentation require detailed formulation of resuspension and deposition. Although other effects of elevated nitrogen could be postulated, the result of this exercise is to recognize that both modelling and measurement streams fall short of ultimate impacts. We are then faced with modelling indirect indications of nutrient enhancement rather than direct changes to the ecosystem. This is a persistent reality in predicting ecosystem health. The essential outcome of this discussion is that we tend toward models of 'indicators of impact' rather than the impact itself. Despite this compromise, we still gain progress toward gauging the extent of the potential influence of aquaculture on the ecosystem.

INDEX MODELS

Aquaculture impact may be defined as waste production which exceeds the capacity of the environment to assimilate, remove or disperse it (one definition of carrying capacity or sustainability). A model or index of impact must therefore include an estimate of waste production as well as an estimate of the removal term. The classic among these indices is related to bivalve feeding, and compares the filtration of seston with its tidal renewal in a simple ratio (Dame, 1996). The same concept has also been applied to production of ammonia (Gillibrand and Turrell, 1997), biodeposits (Grant *et al.*, 2005), and oxygen consumption (Lee *et al.*, 2003). This approach addresses far-field impacts because it averages conditions throughout the whole ecosystem. Yokoyama (2003) devised an embayment degree (ED) to examine the accessibility of fish culture site to tidal exchange. Though not involving culture density explicitly, the ED revisits some of the same scaling arguments in other index models.

Despite the appeal of these approaches, the loss side of the equation is problematic. Loss occurs as a result of diffusion, advection and/or sinking, all of which require a separate estimate or model of physical processes. Sinking is particularly vexing since particles undergo aggregation, disaggregation, deposition and resuspension. These behaviours are poorly known and have the additional complexity of boundary layer physics which regulate their fate. Some models, including DEPOMOD (see section 1), have been successful at making these predictions on a local scale, but prediction of sedimentation rate for a whole bay is a significant task.

The degradation of organic input by the benthos is another loss term, but no less complex. Benthic carbon demand varies as a result of many factors including

temperature, sediment type, and oxygen conditions. One could provide an average estimate of benthic respiration for a bay, but this would require matching to an input term (organic deposition) that is fraught with uncertainty as described above. The ability of the benthos to metabolize organic input is referred to as assimilative capacity (see below).

In order to avoid some of these pitfalls, I have suggested comparing waste production to tidal exchange, by considering the mass of biodeposits prior to deposition (Grant *et al.*, 2005). This approach still provides an index of waste removal but does not require the uncertainty of modelling sedimentation. An analogous example for the dispersion of ammonia is detailed below.

SPATIAL MODELS

I have placed an emphasis on these index models because they are easy to calculate using spreadsheets. With the addition of risk analysis (see below) and a user-friendly interface, these models could be available as desktop tools for habitat managers. This supplements existing tools such as GIS and decision support systems (Hargrave, 2002).

Although these tools are invaluable in site assessment, their suggestion of problems may direct managers to seek more qualitative predictions of impact in the form of simulation models. If a site is to receive continued attention for culture, an increased commitment to predictive capability may be required. I mention box models (e.g. Dowd, 2005 – also see Section 2) as a manageable format of simulation since their computational requirements are small. However, they incorporate spatial delineation, and therefore more development than the spreadsheet format used for index models. Moreover, simple tidal prism calculations will not suffice, and a numerical model of circulation is required to quantify exchange processes in different areas of the bay. These box models are well developed, and approaching the status of 'standard' as far as their trophic components. Once a box model has been properly validated for a given region, it can be used in a desktop environment. For this purpose, we have written a front end to a box model of Magdalen Islands mussel culture (J. Grant, unpublished).

What is to be gained with this increased effort over an index model? Namely, variables such as biodeposition and nutrient concentration may be examined as a function of shellfish stocking density; the range of feasible "what if" including time-dependent scenarios is substantial. Unfortunately, there is considerable spin-up time associated with box models, due to their physical exchange requirements, as well as that of boundary time series. The progression to spatial models from index models may be necessary where there are multiple lease applications in a given bay and/or continued culture development through time. Because index models do not specify space, they cannot be used to consider the interaction between farms. They can however be used to set an overall cap on proposed culture area.

GROUNDTRUTHING

One of the consequences of system-wide indices is that they integrate over space and time and are thus not characterized by any single measurement. Grant *et al.* (2005) utilized sediment trap measurements to verify an index-type model of biodeposition from mussel culture. An important consideration was the extent of spatial and temporal variation in the field observations. Sediment traps are one of the few monitoring tools that can be used to derive rate measurements simultaneously at multiple spatial locations. In contrast, field measurements of water column properties such as dissolved nutrients are biased with tidal, seasonal, and both horizontal and vertical spatial variation. One advantage of index models is that they often involve relative values meant to provide inter-system comparisons without necessity of groundtruthing. Their strength lies in the appropriate scaling. It is a matter of creative uses of spatial and temporal integration to find ways to compare index predictions to field measurements.

EXAMPLE OF AN INDEX MODEL

The premise behind many index models is that a biological term which alters the ecosystem in some way (e.g. waste production, particle consumption) is alleviated by tidal or other exchange mechanisms which remove wastes or renew depleted resources. The production of fecal wastes is derived from knowledge of bioenergetics, i.e. how efficient is an animal at digestion, and therefore how much fecal and dissolved nutrient material is left over. Because the bioenergetics of many cultured animals is well known, a theoretical determination of waste output is reasonably reliable, and in fact easy to obtain. Examples of these calculations may be found in Grant and Bacher (1998), Grant *et al.* (2005), and Gillibrand and Turrell (1997). In order to scale estimates for individual animals up to the entire cultured population, it is necessary to multiply by culture density. The latter term is typical culture density, e.g. for mussels stocking density and longline spacing, diluted by the volume of the entire bay. This value thus incorporates the size of the proposed culture as well as the size of the receiving waters, an essential mode of scaling for a one-box model. More detailed information on proposed culture available from the farm proponent will make this determination more accurate. If desired, adjustments could also be made for animals of different sizes using allometry.

Although previous formulations of index models have used an absolute concentration as output, a non-dimensional expression based on time to reach a given concentration as a function of input has intuitive appeal. The waste production side of the equation may thus be considered in terms of the time required t_M to enhance background values by a given multiple M

$$t_M = M/r$$

where r is the population rate of waste production (units T^{-1}) derived from bioenergetics. The population excretion is not dependent on ambient ammonia concentration, so a simple linear function will suffice. For example, if the level of concern is M= 10x greater waste concentration in the presence of culture, the equation is t_M= 10/r. Because culture density is relative to the size of the entire bay, a consideration of the size of proposed culture relative to size of the bay is explicit in the calculations.

Flushing time (f_t) may be calculated as e-folding time to provide a measure for comparison to nutrient increase time t_M. It is important to identify the primary tidal constituents for calculation of e-folding time (see Grant *et al.*, 2005). Once this flushing time is determined, it can be compared to nutrient addition time as a dimensionless ratio

$$I_A = t_f/t_M$$

Although I will not undertake extensive examples of this ratio, a representative t_M for mussels in dense culture (10 ind m^{-3}) is ~1 day. This density averages over areas of the bay without and without culture, and likely represents a bay with extensive farming. The range of t_f for coastal waters is usually >1 day, and the index is thus >1 for the example above, indicating concern that ammonia concentration will rise. Values of $I_A < 1$ indicate that the exchange of water occurs in a shorter time than the time required for a 10x increase in ammonia over ambient values. Scaling back of culture levels to lower I_A values may alleviate this concern. This example demonstrates that a great deal of information can be incorporated into a simple dimensionless index that gauges whether a planned activity is excessive in terms of ecosystem health. A family of curves can be generated plotting I_A versus culture density for different values of enhancement factor M.

RISK ANALYSIS

One problem with simple models is that they generate single values of the index based on multiple parameters, e.g. bioenergetic calculations. Although one could incorporate sensitivity into some of the parameter estimates, this is tedious by hand. Spreadsheet add-ons specifically tailored for error analysis are better suited to this task. These may be characterized as risk analysis because they formalize the contribution of various model terms to the index estimate, and partition the risk that the index is in error as a function of its parts. This can also be used to gain insight into less rigorous components of the model. We have used @RISK (www.palisade.com) to quantify these error terms. In addition, we have generated a custom Windows-based interface into index model calculations so that parameters such as culture density may be inserted, and the risk analysis automated. Work on this topic continues.

OTHER INDICES

Although it is possible to calculate a variety of indices for ecosystem health (Jorgensen *et al.*, 2005), many of these are retrospective, utilizing empirical data. They are not intended for prediction or incorporation of 'what if' data such as culture density.

Assimilative capacity

Assimilative capacity may be defined as the ability of the water column or benthos to accept organic input without compromising ecosystem health. For example, a slight increase in organic sedimentation will increase benthic production, but further inputs will drive the system to oxygen stress. This relationship implies a hyperbolic relationship between sediment oxygen demand (a measure of activity) and organic input. There have been efforts to define this relationship and apply it as a management tool for aquaculture (Yokoyama, 2003), but there are many factors that define the curve, and practical application has proved difficult. There is an inherent appeal of benthic oxygen demand since it is a direct measure of carbon consumption and thus waste removal. There are many measurements of this process at aquaculture sites, and assimilative capacity should remain an area of research especially as it relates to key monitoring variables such as redox and sulfide. Other integrative measures designed to incorporate aspects of organic loading such as the Benthic Enrichment Index (BEI; Hargrave *et al.*, 1997) are intended to utilize site-reference comparisons based on empirical data. They are less suited to prediction.

Seston depletion

Seston depletion compared to renewal is the classic predictive index applied to cultured and natural populations of bivalves. It is appealing because both renewal and filtration are expressed as turnover time, with no need for thresholds. Seston depletion has also been considered in more local models of farm scale filtration. Unfortunately it is difficult to relate depletion to other ecological processes. At the system scale, excess filtration is presumably making the water 'too clean' without clear implications for other grazers or trophic levels. Among indices, the depletion index has been compared most broadly among systems (Prins and Dame, 1998). The historical use and ease of calculation argue for inclusion in a standard set of tools useful for shellfish culture.

Oxygen consumption

Oxygen is not necessarily limiting in marine waters except under conditions of poor flushing, high consumption, and high temperatures. There are naturally hypoxic waters arising from stratification, creating a potentially complex interaction with aquaculture sites where oxygen demand of the water column and benthos is elevated. It is feasible to calculate a predictive index analogous to the ammonia example detailed above, as Lee *et al.* (2003) did for tropical cage culture. For temperate waters, one might suggest that other effects such as sediment anoxia

from biodeposition become important prior to water column oxygen stress, making the latter a less sensitive predictor of problems.

Trophic indices

Trophic structure may be altered by differential species or functional group response to organic enrichment. The best known example is a shift to opportunistic species in the benthos due to eutrophication or other disturbances (Newell *et al.*, 1998). Trophic indices based on taxonomic delineation are expensive to monitor, and have only occasionally been the subject of modelling (Duplisea, 1998). Except under extreme impacts, changes in benthic communities may be highly localized and less responsive as system-wide variables. The relationship between organic carbon loading and benthic biodiversity has been proposed via the infaunal tropic index incorporated into DEPOMOD (see Section 1), but this is again a local-scale application.

At lower trophic levels, alteration of nutrient and particle budgets via processes such as animal excretion and feeding might be expected to cause changes in phytoplankton populations or the microbial loop. For example, Harrison *et al.* (2005) suggested that the ratio of chlorophyll to bacterial abundance allowed distinction of far-field impacts at a shellfish aquaculture site. This index demonstrates that some indices are based on concentrations of certain variables as opposed to rate processes. Similarly, Meeuwig *et al.* (1998) applied index models to a whole range of estuaries in PEI using chlorophyll:nutrient values to assess the impact of mussel culture. The approach entailed determining the extent of nutrients, turbidity and grazing in contributing to chlorophyll levels in these estuaries. Nonetheless, their study demonstrates the extent to which index models can be applied to ecosystem-level characterization.

USE OF MODELS IN CULTURE SITE ASSESSMENT.

Based on the above considerations, a suggestion for how simple models may be incorporated into aquaculture management is proposed:

1. Establish a GIS for the bay, including digital boundaries and bathymetry, and any available layers. The extent of GIS expertise at both federal and provincial levels makes this task entirely feasible. This base level of information is considered essential for any management scenario.
2. Obtain a tidal time series for the bay of interest, e.g. using free tidal database software (www.wxtide32.com). Use this information and bay volume to calculate a baywide flushing time. Gregory *et al.* (1993) have done this for many bays and estuaries in the Maritimes.
3. Use detailed plans of the culture site to estimate animal density.
4. Use the above information to calculate selected index models for the proposed site.

5. Adjust the proposed culture level until the index assumes acceptable values. Use this value as a guide for setting an upper limit to culture in the bay.
6. For index values exceeding this limit, consider further modelling for spatial predictions.

These steps assume that that appropriate index models have been selected and that a scoring system for prediction has been established. It is highly unlikely that a small lease will yield index values of concern. More often, values exceeding a comfort threshold will occur in bays with widespread culture or plans for continued expansion. If index models indicate that there is increased potential for impacts, then further modelling should be undertaken, perhaps with multi-box ecosystem models. This increased effort is necessary in bays that will remain a focus for aquaculture. Because the spatial models have so much potential for bay management in activities well beyond aquaculture, they are a valuable investment for designated bays. More importantly, the underlying circulation model is an essential tool for similar reasons. Fisheries and Oceans Canada has ample capability for generation of both types of models and should foster their inclusion as a regular part of the assessment process where ongoing management will be required beyond an isolated farm.

WORK TO BE DONE

Index models have an intuitive appeal in that the comparisons they engender are easy to understand and relay to the public. Although there is some work to be done, it's not especially onerous. A survey of various indices should be undertaken to designate the most robust examples in terms of predictive capability, ease of calculation, sources of error, and data requirements. There are sufficient data at many sites to undertake these comparisons. Although I have emphasized indices which contrast waste production versus tidal renewal, other indices such as those in Meeuwig *et al.* (1998) or Yokoyama (2003) should be examined. A combination of indices is the way to go, with the incorporation of risk analysis and desktop interfaces. Although there is always room for improvement (e.g. further statistical properties), index models can be used for aquaculture management immediately, with ongoing research providing further refinement of their application to aquaculture as well as other issues in the coastal zone. The intuitive nature of scaling arguments which specifically address the question of 'how much is too much?' constitute an important tool in the management of far-field impacts based on a fundamental understanding of the ecosystem.

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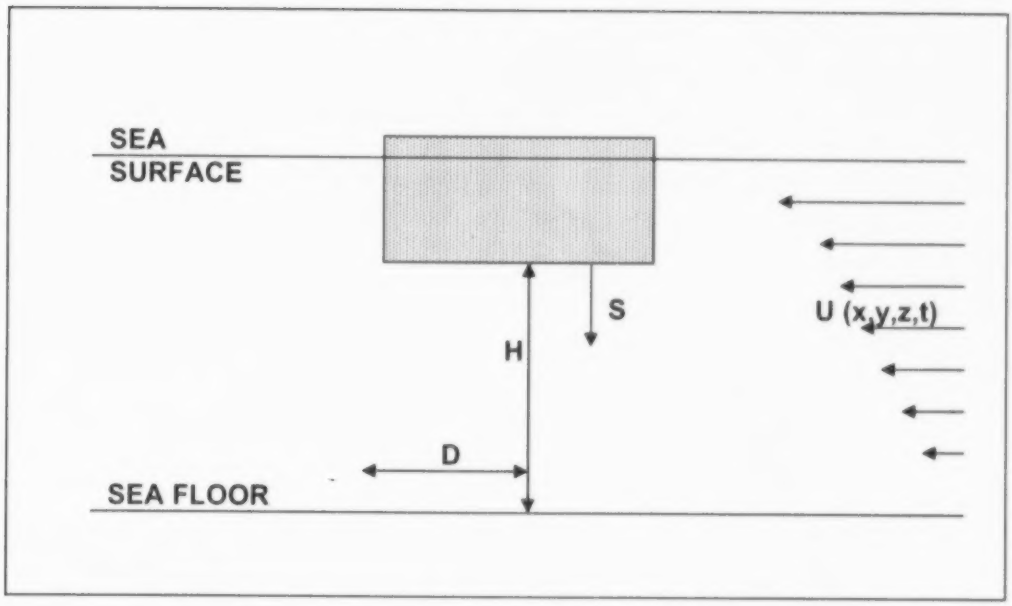

Figure 1. Schematic diagram showing the components and main processes relating to the dispersion and transport of solids wastes from aquaculture operations. The capital letters, S, H, and U respectively refer to the sinking rate and vertical fall distance for the particles, and the spatially and temporally varying horizontal current velocity.

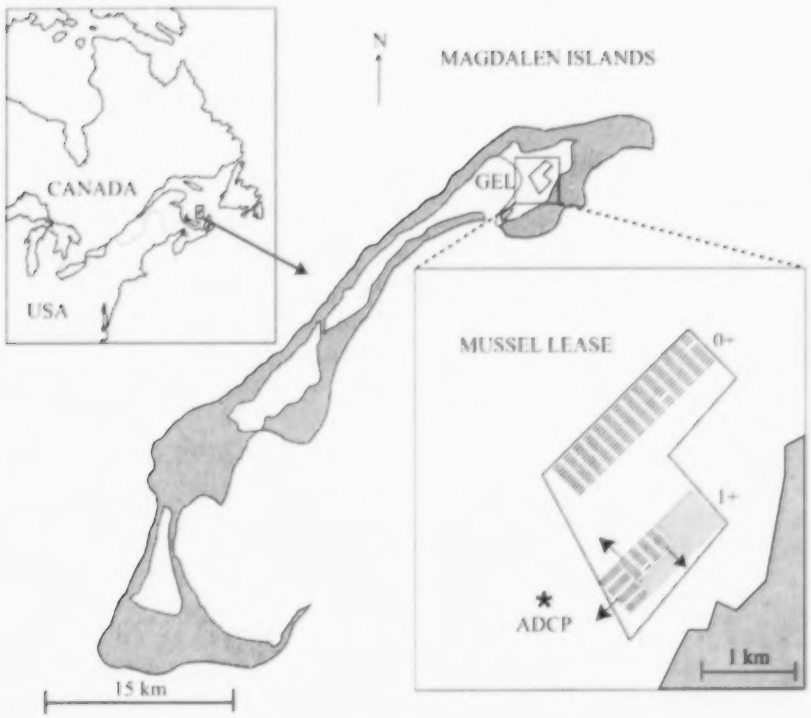

Figure 2. Map of the Magdalen Islands, Quebec, showing the location of the mussel farm. The farm is divided into two zones based on age classes: 0+ and 1+. Mussel lines are indicated by black dashed lines (-) and the area of harvested mussel lines (1+ zone) is in light grey. The black arrows represent transects, which ran perpendicular to the last mussel line on each side of the 1+ zone. The asterisk (*) indicates the position of the ADCP. This figure has been modified from Callier *et al.* (2006).

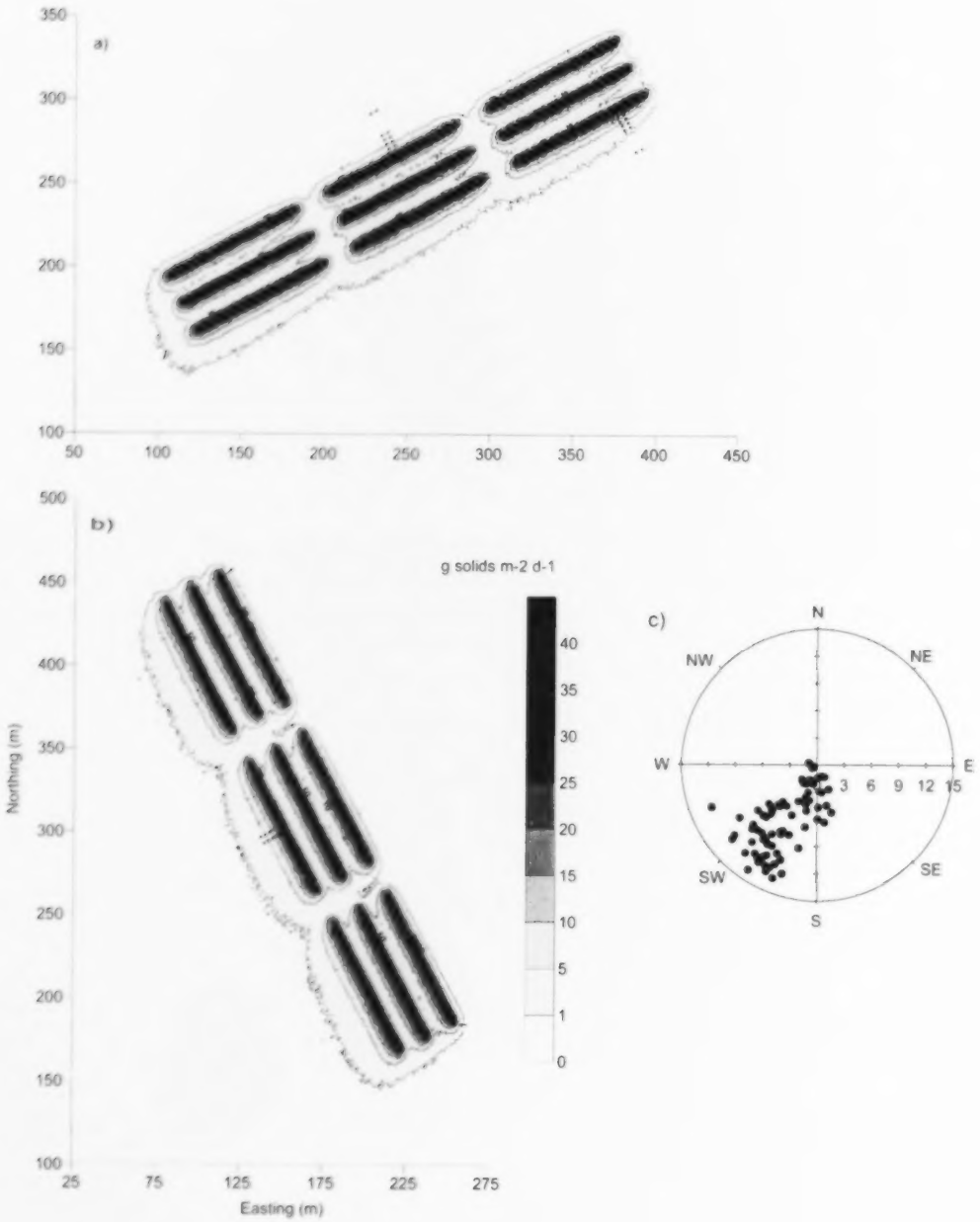

Figure 3. The modelled flux (g m^{-2} d^{-1}) of solids accumulation on the seabed arising from a series of mussel lines. The sediment trap positions (■) are shown for each site: a) transects towards the NW and SE, b) transect towards the SW. Paired sediment traps, separated by 4 m, were positioned at distances of 0, 3, 6, 12, 15 and 30 m, perpendicular to the last mussel line on three sides of the mussel site. The current direction and speed (cm s^{-1}) recorded every 20 min during the 24 h sampling period is illustrated in c).

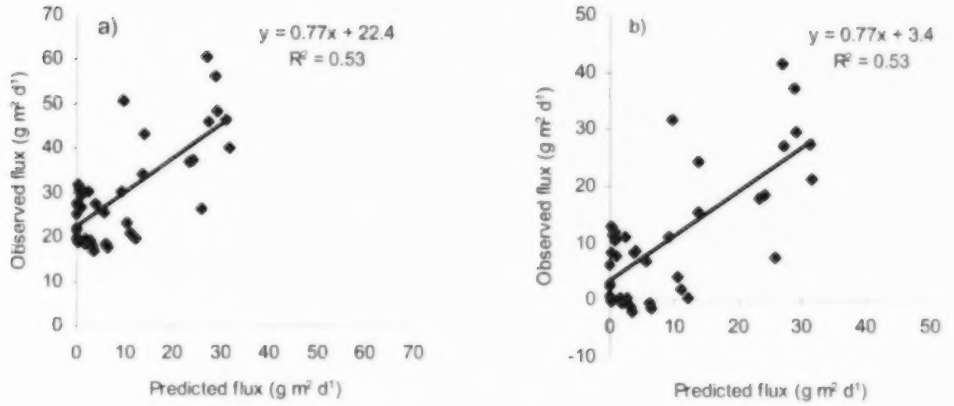

Figure 4. Comparison of observed and modelled solids deposition (n = 38) over a 24 h period beneath and around the mussel farm in Great-Entry lagoon. The predicted flux is compared to observed sedimentation rates in a). A background sedimentation value of 19 ± 5 g m^2 d^1 was subtracted from observations in b) based on data collected from 60 sediment traps deployed on four dates at the time of the study (Callier *et al.*, 2006).

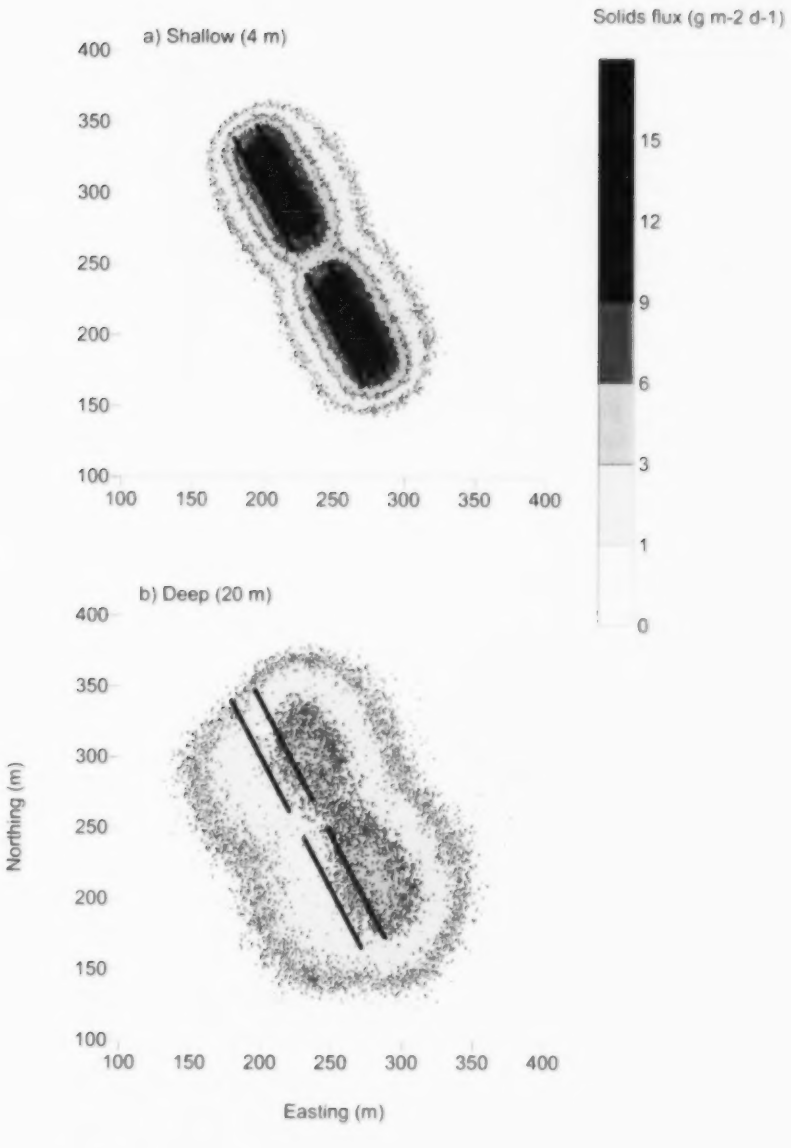

Figure 5. Scenario 1 – Shallow site versus deep site. Model predictions of flux (g m^{-2} d^{-1}) showing the footprint area around four mussel lines (shown as lines). The flux (impact) is greater at the shallower site (dark areas). Although the deeper site shows a lower flux beneath the mussel lines, biodeposits are dispersed more widely.

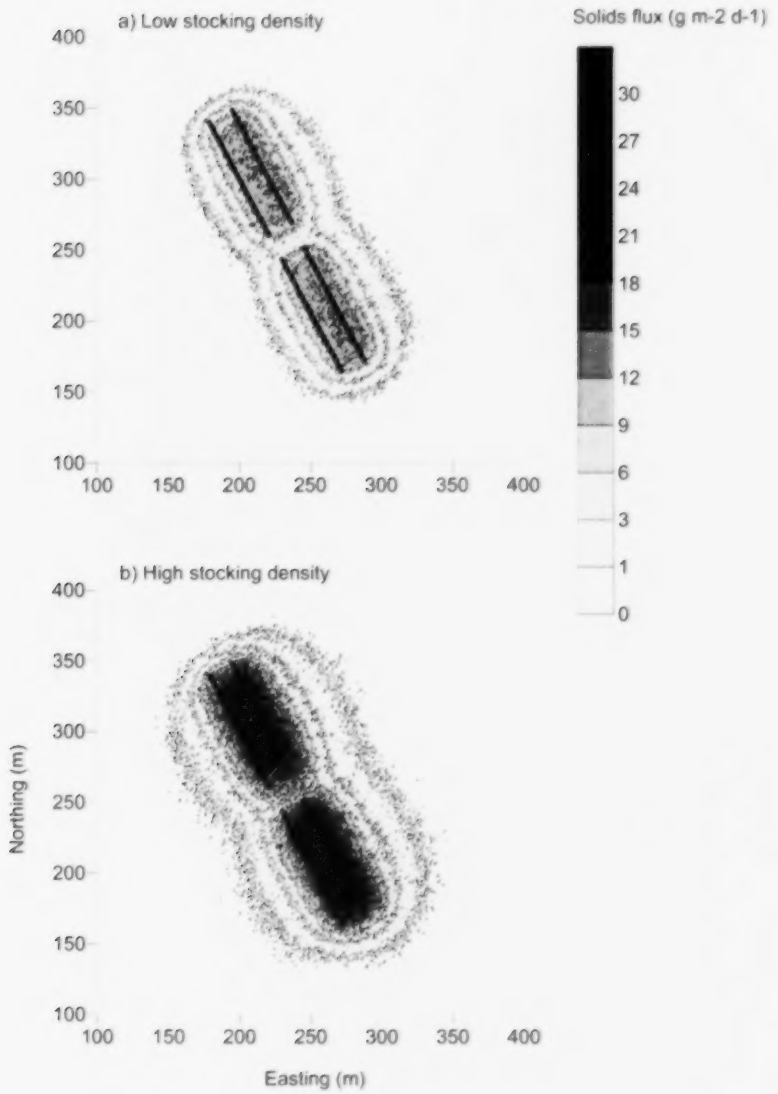

Figure 6. Scenario 2 – High and low stocking density. A feed input of 26.4 kg cage^{-1} d^{-1} was used for the low stocking density in a) and doubled to 52.8 kg cage^{-1} d^{-1} in b). The model predictions of flux (g m^{-2} d^{-1}) show a higher level of impact underneath the mussel lines with high stocking density.

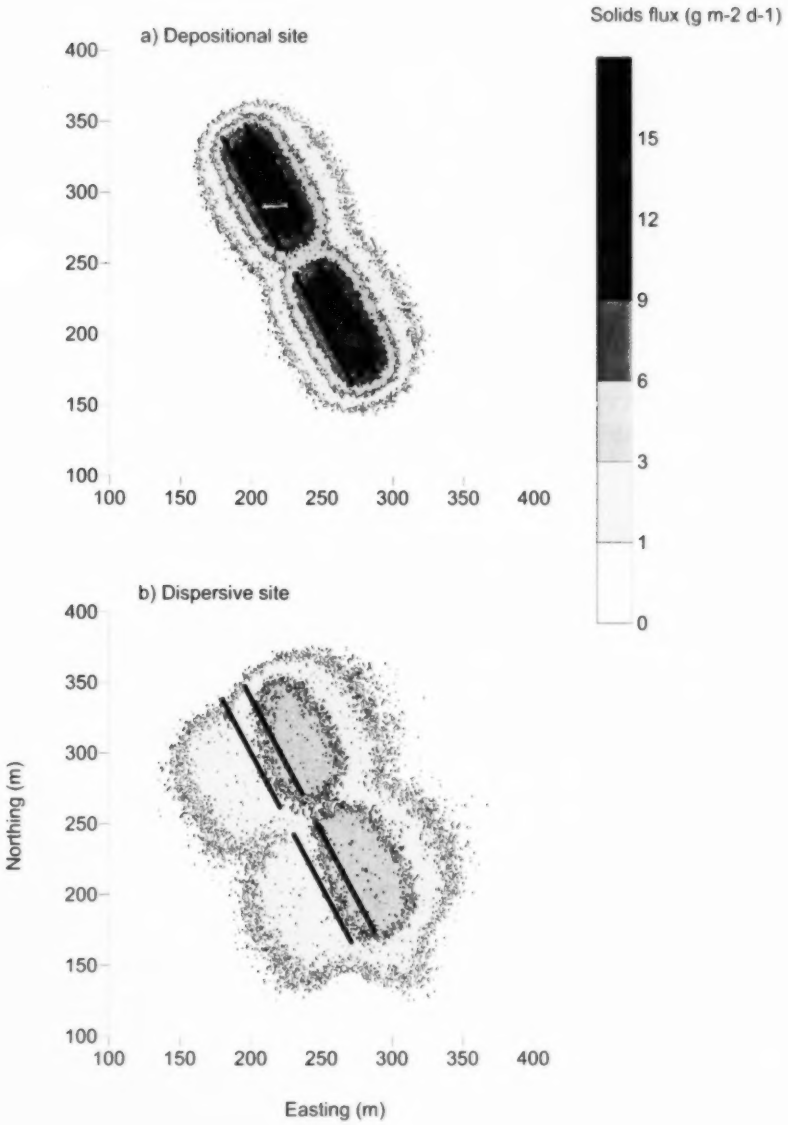

Figure 7. Scenario 3 – Low and high current velocities. Model predictions of flux (g m^{-2} d^{-1}) around mussel lines located in a) a depositional and b) dispersive site.

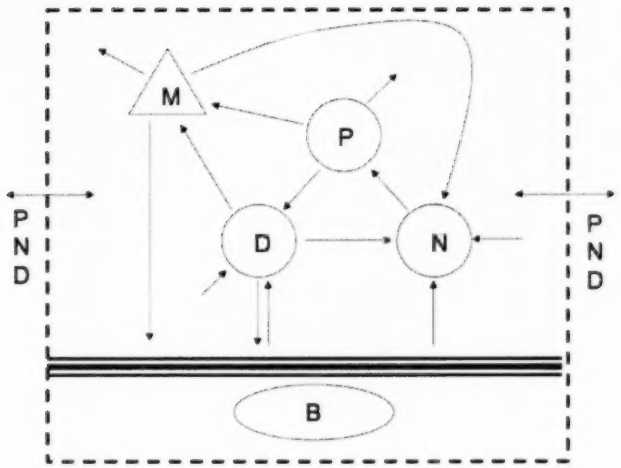

Figure 8. Schematic diagram of a simple lower trophic level ecosystem box model which includes bivalve aquaculture. The pelagic state variables are phytoplankton *P*, nutrients *N*, and detritus *D*. A benthic detrital pool, *B*, is included. Bivalves, *M*, are incorporated as a diagnostic variable. Arrows represent ecosystem fluxes between components and with adjacent regions.

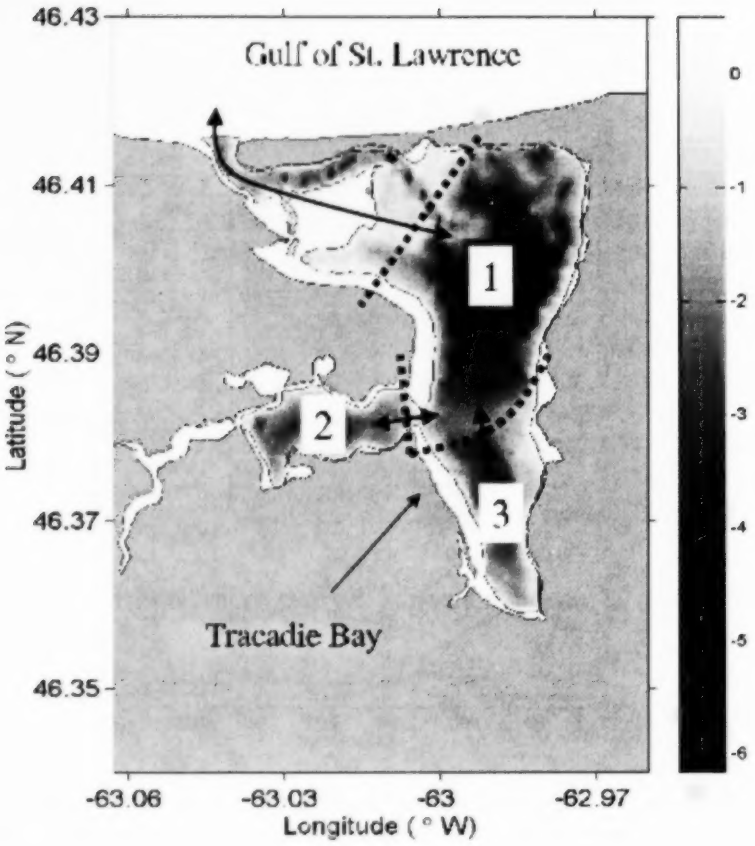

Figure 9. Map of Tracadie Bay, PEI showing its division into three boxes. Dotted lines represent box boundaries and arrows indicate exchange of pelagic ecosystem components between the boxes and with the offshore (far-field).

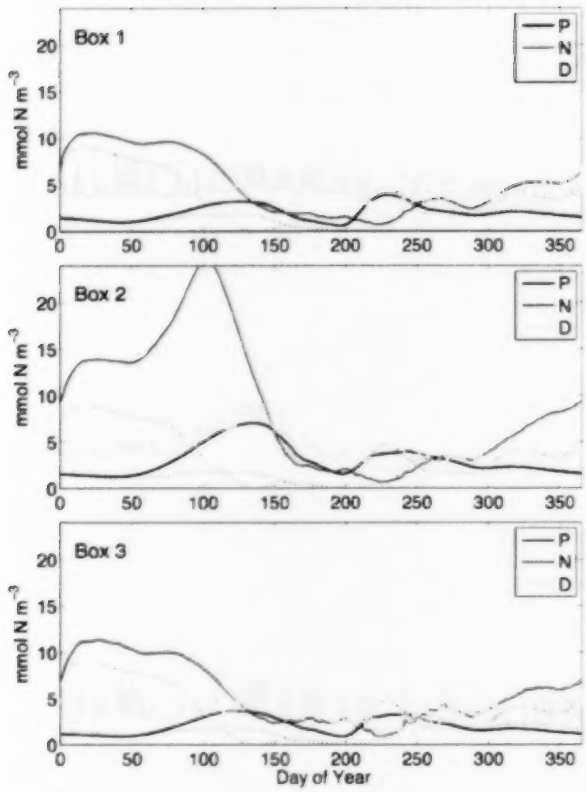

Figure 10. Simulated annual cycle of pelagic ecosystem components *P*, *N*, and *D* in Tracadie Bay for each of the three boxes. Dotted lines indicate far-field values.

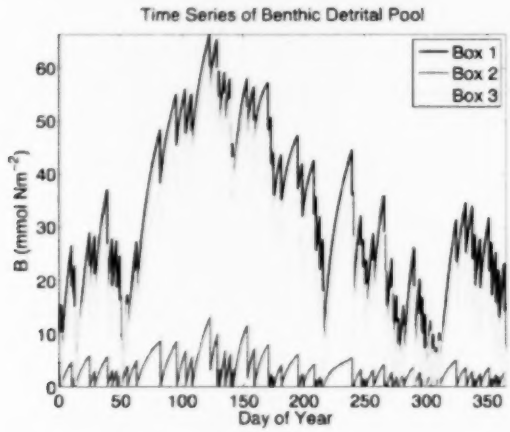

Figure 11. Simulated annual cycle of the benthic detrital pool, *B*, in Tracadie Bay for each of the three boxes.

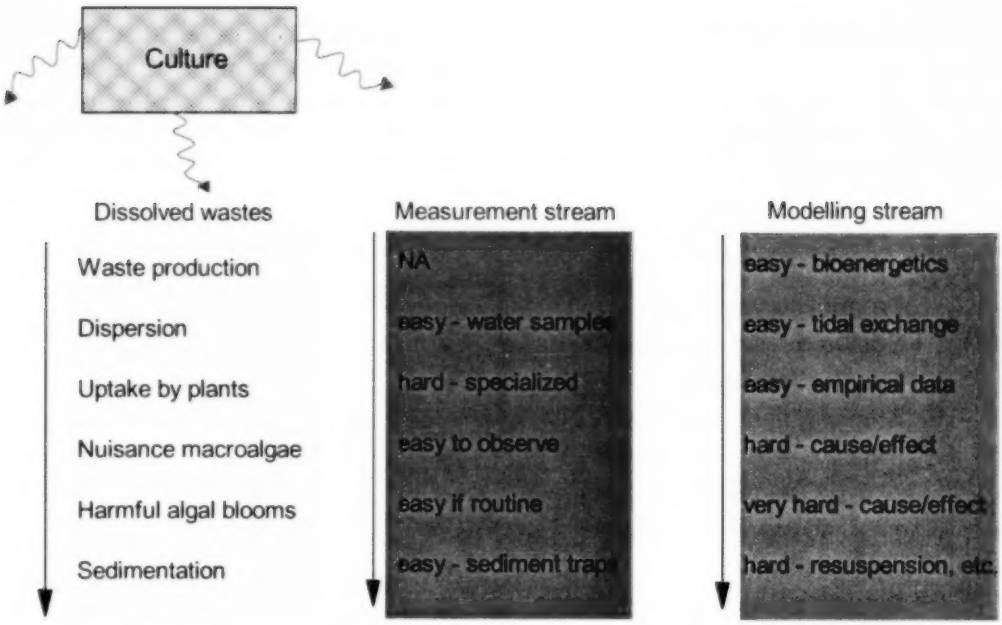

Figure 12. Comparison of dissolved nutrient effects to measurement and modelling capability.

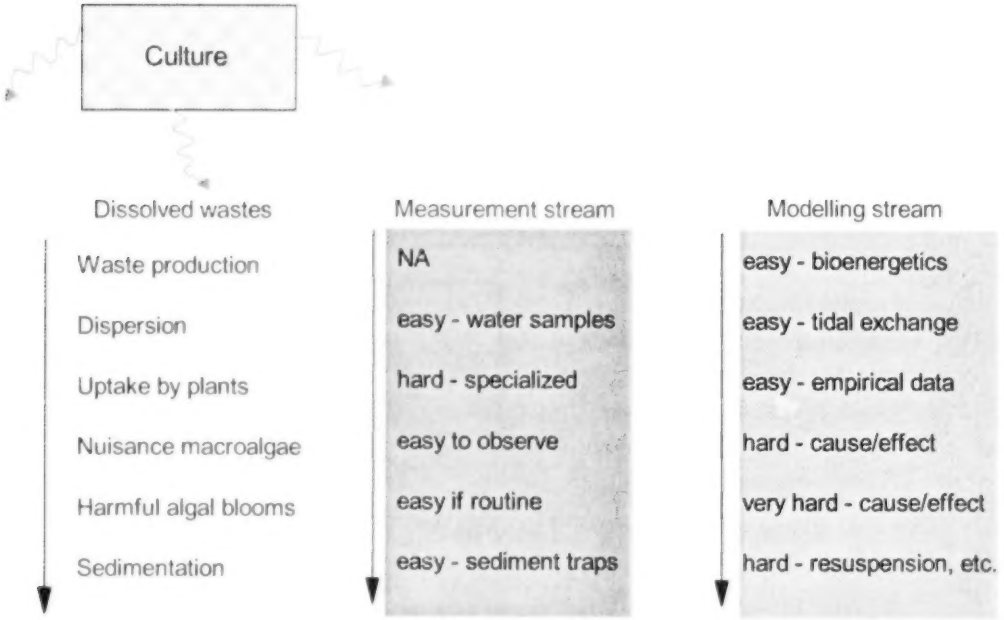

Figure 12. Comparison of dissolved nutrient effects to measurement and modelling capability.

Table 1. Factors that may affect the nature and scale of effect of suspended shellfish aquaculture operations on the proximal benthic environment. These are classed into groups characterizing how they influence the potential effect; A - quantity and quality of material exiting the farm; B - dispersion of material exiting the exiting the farm; and C - fate of waste material post-deposition.

Factor	Group
Production tonnage	A
Seston availability to stock	A
Feeding behaviour	A
Stocking density on ropes	A
Physical structure of farm	B
Orientation of ropes	B
Distance to seabed	B
Hydrodynamics	B
Settling velocity of biodeposits	C
Resuspension	C
Degradation	C
Removal (eg consumption)	C
Age of farm	C

Table 2. Input data used for the grid generation and particle tracking modules for the sediment trap study (model validation).

Input data	GEL 2003
Grid size (m)	499 x 499
Grid cell resolution (m)	1 x 1
Number of mussel lines	3 x 3
Cage dimensions (l x w x d) (m)	91 x 0.2 x 1
Distance between mussel lines (m)	18
Cage orientation (deg)	NW-SE or SW-NE
Depth under mussel lines (m)	0-1.5
Grid bathymetry (m)	flat
Number of current velocity data sets used	1
Length of current velocity record used (h)	24
Instrument sampling period*	2 min in every 20
Feed input (kg d^{-1} $cage^{-1}$)**	26.4
Number of particles used	3×10^6
Particle trajectory evaluation (s)	6
Sinking speed of faeces (cm s^{-1}), 67% of particles	1.0 ± 0.3
Sinking speed of pseudofaeces (cm s^{-1}), 33% of particles	0.2 ± 0.02
Horizontal diffusion coefficients (k_x, k_y) (m^2 s^{-1})	0.1
Vertical diffusion coefficients (k_z) (m^2 s^{-1})	0.001

*A 500 kHz ADCP sampled in pulse-coherent mode in 20 equally spaced cells of 0.25 m thickness from 0.6 to 5.6 m above the sediment bottom.

**The "feed input" for the 1+ mussel cohort is based on 575 mussel m^{-1}, a biodeposit production rate of 125.6 mg d^{-1} $mussel^{-1}$ and a length of 365.8 m per "cage".

Table 3. Annual fluxes between ecosystem components and the marine environment in kilomoles of Nitrogen per year. Two scenarios are shown: (i) bivalve aquaculture at present level, and (ii) no bivalve culture. The third column represents the difference between these two scenarios. The notation $X \rightarrow Y$ refers to a flux from ecosystem state variable X to Y. Inputs and outputs to the system are also given.

	M Present	M Absent	Difference
$N \rightarrow P$	8539	8907	-368
$P \rightarrow D$	3869	5119	-1250
$D \rightarrow N$	5726	6402	-676
$B \rightarrow N$	4205	602	3603
$D \rightarrow B$	2832	3232	-400
$B \rightarrow D$	1537	629	908
$P \rightarrow M$	3453	0	3453
$D \rightarrow M$	6545	0	6545
$M \rightarrow B$	8348	0	8348
$M \rightarrow N$	1100	0	1100
P export	-1217	-3788	2571
N export	-9726	-5333	-4393
D export	5084	1993	3091
burial	3364	481	2883
N input	7235	7235	0
M harvest	550	0	550